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Recovery of floral and faunal communities after placement of dredged material on seagrasses in Laguna Madre, Texas

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Abstract

The objectives of this project were to determine how long alterations in habitat characteristics and use by fishery and forage organisms were detectable at dredged material placement sites in Laguna Madre, Texas. Water, sediment, seagrass, benthos, and nekton characteristics were measured and compared among newly deposited sediments and nearby and distant seagrasses each fall and spring over three years. Over this period, 75% of the estimated total surface area of the original deposits was either re-vegetated by seagrass or dispersed by winds and currents. Differences in water and sediment characteristics among habitat types were mostly detected early in the study. There were signs of steady seagrass re-colonization in the latter half of the study period, and mean seagrass coverage of deposits had reached 48% approximately three years after dredging. Clovergrass *Halophila engelmannii* was the initial colonist, but shoalgrass *Halodule wrightii* predominated after about one year. Densities of annelids and non-decapod crustaceans were generally significantly greater in close and distant seagrass habitats than in dredged material habitat, whereas densities of molluscs were not significantly related to habitat type. Nekton (fish and decapod) densities were almost always significantly greater in the two seagrass habitats than in dredged material deposits. Benthos and nekton communities in dredged material deposits were distinct from those in seagrass habitats. Recovery from dredged material placement was nearly complete for water column and sediment components after 1.5–3 years, but recovery of seagrasses, benthos, and nekton was predicted to take 4–8 years. The current 2 to 5 year dredging cycle virtually insures no time for ecosystem recovery before being disturbed again. The only way to ensure permanent protection of the high primary and secondary productivity of seagrass beds in Laguna Madre from acute and chronic effects of maintenance dredging, while ensuring navigation capability, is to remove dredged materials from the shallow waters of the ecosystem.

Keywords: water column; seagrass; sediments; benthos; fishes; decapods; dredging; community; recovery

1. Introduction

Many of the world's coastal embayments are dredged for shipping and navigation purposes. The Gulf Intracoastal Waterway (GIWW) was dredged through the Laguna Madre of Texas during 1945–1949 by the U.S. Army Corps of Engineers (Rickner, 1979; Brown and Kraus, 1996), and it forms the only permanent connection between Upper and Lower Laguna Madre. Sediment from original channel deepening was deposited alongside the GIWW in placement areas that continue to be used for maintenance dredging on a 2 to 5 year cycle (Chaney et al., 1978). Much of the Laguna Madre bottom is covered by seagrasses (Quammen and Onuf, 1993; Onuf, 1994, 1996), and the estuary supports diverse communities of fishes and invertebrates that include many economically valuable species (McEachron and Fuls, 1996; Tunnell et al., 1996; Robinson et al., 1997). Dredged material deposits may have acute and chronic effects on seagrass habitats, their use by fishery and forage organisms, and overall system productivity. Periodic placement of dredged material may reduce the ecological value of the bottom through direct effects (e.g., covering seagrasses or mudflats, smothering benthos, increasing turbidity, or exchanging one habitat [seagrass] for another [subtidal mudflat or emergent island]) or indirect effects (e.g., altering prey densities

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and compositions, or altering predation rates by reducing water clarity).

Disposal of dredged materials causes immediate, widespread, but usually short-term, effects on water clarity (Windom, 1975; Nichols et al., 1990). In Lower Laguna Madre, however, dredging deposits led to elevated turbidity on-site for up to 15 months after deposition, and increased turbidity was noted in adjacent seagrass beds up to 10 months after deposition and at distances >1.2 km away from placement areas (Onuf, 1994).

Placement areas can be re-colonized by seagrasses, although time frames are variable and re-colonization is not assured. A seed-set of eelgrass *Zostera marina* on a previously non-vegetated placement area in Core Sound, North Carolina, led to shoot densities and coverage similar to those at the edge of adjacent natural beds within six months (Fonseca et al., 1990). Shoalgrass *Halodule wrightii* advanced 3–4 m in less than two months onto a placement area in Upper Laguna Madre that had remained non-vegetated for the previous 12 years (Circé, 1979). In Redfish Bay (just north of Upper Laguna Madre), seagrasses began to colonize the edges of a deposit only after three years (Hellier and Kornicker, 1962; Odum, 1963). In Lower Laguna Madre, Sheridan and Minello (2003) found no seagrass colonization of dredged material at two sites after three years, even with a transplanting effort, because placement sites were subject to relatively strong currents and sediments were apparently unstable. Where seagrasses have colonized dredged material, reduced seagrass shoot densities persisted for 10 years in Upper Laguna Madre (Rickner, 1979) and for 31 years in Indian River Lagoon, Florida (Brown-Peterson et al., 1993).

Benthic community recovery after dredged material deposition has not been well studied, particularly with reference to seagrass habitats. Placement areas in Coos Bay, Oregon, that experienced chronic disturbance from passing ship wakes exhibited rapid re-colonization by benthos (within seven days), presumably because the community was adapted to disturbance (McCauley et al., 1977). An open water disposal site in Long Island Sound, New York, supported densities of annelids similar to those in control areas within seven months, although mollusc densities had not reached parity after 12 months (Rhoads et al., 1977). A shallow (1 m) subtidal sand deposit in Apalachee Bay, Florida, attracted high densities of annelids and crustaceans within three months, but the benthic assemblage did not resemble that of nearby seagrass beds after 12 months (Subrahmanyam, 1984). In Upper Laguna Madre, Rickner (1979) reported that mollusc and polychaete species compositions and densities in seagrasses that had colonized dredging deposits required at least 10 years to become similar to communities in adjacent natural seagrass beds.

Nekton communities (fishes and decapod crustaceans) can respond quickly to dredged material deposits, but community alterations persist through time. Repeated comparisons of non-vegetated dredged material and seagrass habitats 1.5–3 years after deposition in Lower Laguna Madre indicated that fish densities did not differ significantly, but fish community compositions were distinctive (Sheridan and Minello, 2003). The reverse was found for decapods in that study—densities were always significantly lower at dredged material sites but species compositions were similar to those in seagrasses. Once dredged material deposits were thoroughly colonized by seagrasses, however, there were no differences in nekton communities between vegetated placement areas and natural seagrass beds (Sheridan and Minello, 2003). Elsewhere, Fonseca et al. (1990) offered a single observation that a six-month-old, naturally re-seeded eelgrass bed on a disposal site contained densities of fishes and shrimps similar to those in the edge of nearby natural beds in Core Sound, North Carolina. Brown-Peterson et al. (1993) compared fish communities of natural seagrass beds and colonized beds 31 years after dredging of the Atlantic Intracoastal Waterway in Indian River Lagoon, Florida. Persistent differences in fish species composition were noted, even though fish densities were similar during most seasons. Thus, re-establishment of seagrass in an area does not assure the re-establishment of habitat value for fishes and decapods.

The objectives of this project were to determine how long alterations in habitat characteristics and in habitat use by fishery and forage organisms were detectable at placement sites in Laguna Madre, Texas, and whether impacts persisted over a typical dredging cycle. Maintenance dredged material that was placed during early 1995 was monitored at three sites each in Upper and Lower Laguna Madre. At each site, three habitats were examined: (1) dredged material deposits that buried seagrasses; (2) seagrasses within 5 m of the deposits that received little fresh dredged material but were expected to experience greater turbidity and other indirect effects of material placement; and (3) undisturbed seagrasses about 1 km away. Seasonal water column parameters, surface sediment characteristics, seagrass biomass and coverage, and benthos and nekton densities and species compositions were measured and compared each fall and spring over a three-year period (1995–1998). Certain sections of the Laguna Madre system, particularly Upper Laguna Madre, experienced periodic blooms of the brown tide alga *Aureoumbra lagunensis* during the study period. At the time, it was unknown what effects brown tide might have on re-colonization of dredged material (Onuf, 1996; Street et al., 1997).

2. Methods

Laguna Madre is a Texas barrier island estuary extending 200 km from Corpus Christi Bay south to the

Rio Grande delta on the United States–Mexico border. The estuary is separated into Upper Laguna Madre and Lower Laguna Madre by a large depositional fan known as the Land Cut, which forms wind-driven tidal flats in the middle of the estuary (White et al., 1989).

2.1. Site selection

Maintenance dredged material was deposited in 13 placement areas in Lower Laguna Madre and nine in Upper Laguna Madre during late 1994 and early 1995. All 22 sites were visited in August 1995 to determine whether placement areas were subtidal (some became emergent), whether new materials could be located, and whether placement areas supported seagrasses nearby and at a distance of 1 km. Only six placement areas (U. S. Army Corps of Engineers identification numbers 187, 194, 197, 211, 221, and 222) met these requirements. The six sites spanned 20 km of Upper Laguna Madre and 30 km of Lower Laguna Madre (Fig. 1) and were located as follows: 187 at 27° 27.08′ N, 97° 20.00′ W; 194 at 27° 20.85′ N, 97° 22.95′ W; 197 at 27° 17.18′ N, 97° 24.32′ W; 211 at 26° 47.53′ N, 97° 28.01′ W; 221 at 26° 30.71′ N, 97° 28.31′ W; and 222 at 26° 29.00′ N, 97° 23.02′ W. Positions were noted with a global positioning system (Garmin International, Model 38, Olathe, Kansas; accuracy 100 m).

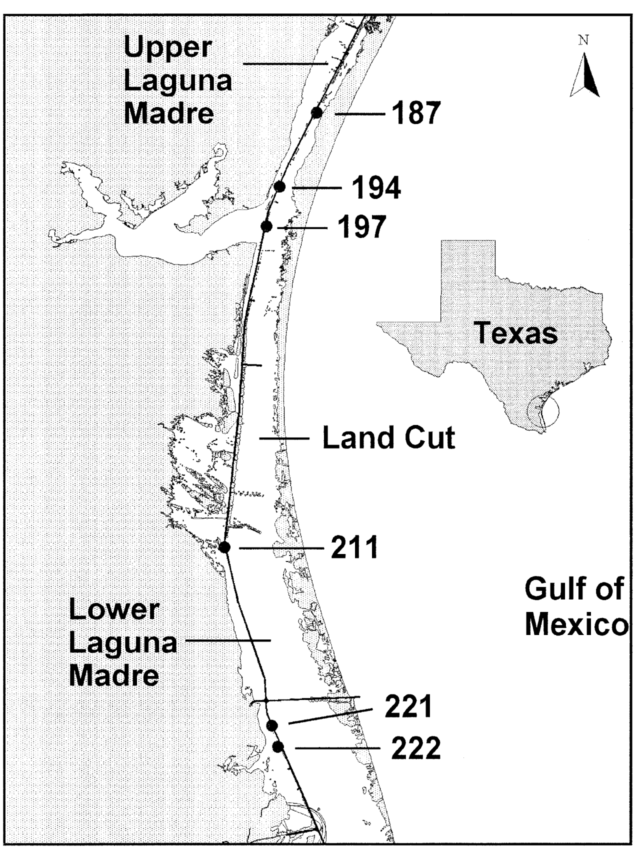

Fig. 1. Map of the study area on the south Texas coast. Dredged material placement areas (3 digit numbers) were located immediately adjacent to the Gulf Intracoastal Waterway.

Dredged material was placed on 187, 194, and 197 in January 1995 and 211, 221, and 222 in March 1995. Projected volumes by placement area were: 187 = 77,420 m^3, 194 = 127,830 m^3, 197 = 160,650 m^3, 211 = 130,050 m^3, 221 = 95,625 m^3, and 222 = 123,930 m^3 (areal coverage was not measured; T. Roberts and N. McLellan, U. S. Army Corps of Engineers, Galveston, Texas, pers. comm.). Each of these placement areas had fringing seagrasses that partially or completely surrounded the site. The closest undisturbed seagrass beds that were not near non-target placement areas and were of similar depth ranges were all located west of the selected placement areas. Brown-Peterson et al. (1993) noted that having all sampling sites located in one half of an estuarine system could affect interpretation of results. However, this was not expected to influence floral and faunal densities in our undisturbed seagrass habitat since Rickner (1979) indicated that floral and benthos densities were unrelated to lateral position within the estuary.

2.2. Experimental design

Sampling was designed to test whether habitat use patterns of flora and fauna were directly affected by dredged material placement and to estimate recovery to the point that placement sites were statistically indistinguishable from nearby undisturbed seagrasses. Sample sets were collected six times over three years: September 1995, April and September 1996, May and September 1997, and April 1998. Sampling was scheduled for fall and spring seasons because fish and decapod abundances in general, and those of fishery species in particular, were expected to be relatively high (Hellier, 1962; Hoese and Jones, 1963; Stokes, 1974; Huh, 1984). Benthic community densities were expected to be intermediate to low (Huh and Kitting, 1985; Montagna and Kalke, 1992).

Power analyses (Sokal and Rohlf, 1981) of fish, decapod, and benthos densities elsewhere in Texas seagrasses (Rozas and Minello, 1998; Sheridan et al., 2003; Sheridan, unpublished data) indicated that 30 samples per habitat type would allow detection of a 100% difference in means between any two seagrass habitat types at $\alpha = 0.05$ and $1 - \beta(\text{power}) = 0.80$. Three habitat types were designated at each of the six sites: Maximum Impact (subtidal dredged material near the center of the placement area, devoid of seagrasses through most of the study); Minimum Impact (seagrasses within 5 m of the edge of dredged material, where turbidity was likely to remain high but seagrasses were still growing); and undisturbed Natural Seagrass (beds approximately 1 km away from each site). Five haphazardly placed sample sets (described below) were taken within each habitat type at each site, yielding the required 30 samples per habitat. Length and width of Maximum Impact habitats were measured by range finder (Ranging Inc., Model 1200, Rochester, NY; accuracy = 1.8 m at 91 m) during each site visit to give a qualitative estimate of non-vegetated dredged material surface area remaining (assumed to be an ellipse of area = $\pi \times$length/2$\times$width/2).

Sample sets included the following types of quantitative data collected from within the perimeter of, or immediately adjacent to, a drop trap used to sample nekton (described below). Temperature, salinity, depth, and turbidity were measured within the drop trap before any other activities (there were no turbidity samples from Lower Laguna Madre during April 1996). Temperature was measured with a YSI Model 55 meter (YSI Inc., Yellow Springs, Ohio). Salinity was measured with a temperature-compensated refractometer. Minimum and maximum water depths were measured with a meter stick. Water samples for turbidity analysis were collected in screw cap bottles and later tested in the laboratory with an HF Scientific Model DRT100B turbidimeter (HF Scientific, Ft. Myers, Florida). Surface sediments (top 5 cm) were collected with a 5-cm diameter corer to examine organic content (loss on ignition; Dean, 1974) and grain size (wet sieve and pipette methods; Folk, 1980). Organic materials were not removed prior to grain size analyses as is usual (Folk, 1980), therefore the terms "rubble", "sand", "silt", and "clay" used hereafter actually refer to rubble-sized, sand-sized, silt-sized, and clay-sized particles. Exclusive grain size limits were: rubble >2 mm, sand >0.0625 mm, silt >0.0039 mm, and clay <0.0040 mm (Folk, 1980). Rubble was tabulated but not analyzed further since it consisted only of shell and seagrass fragments. Seagrass coverage was determined with a 1 m^2 quadrat divided into 20 cm $\times$ 20 cm grid cells following methods of Fonseca et al. (1987) and Dunton (1990). Coverage was derived from the presence or absence of at least one live shoot in a grid cell. Seagrass shoot and root/rhizome biomasses (g dry weight; all species combined) were measured from benthic cores (described below). The root:shoot ratio (RSR) was calculated as an indicator of seagrass carbon storage capacity that is susceptible to stress (Lee and Dunton, 1997). Densities of benthic infauna and epifauna (excluding decapod crustaceans) inhabiting the top 10 cm of sediments and any enclosed seagrass shoots were estimated with a 5.08-cm diameter corer. Three cores per site were pooled (total = 60.8 cm^2) and passed through a 0.500-mm mesh sieve, with the remainder being preserved in 10% formalin plus rose bengal (Sheridan and Livingston, 1983). Densities of nekton (fishes and decapod crustaceans) were estimated with a 1 m^2 drop trap deployed from the bow of a boat during daylight hours (Zimmerman et al., 1984). Research in Florida seagrasses indicated no nekton community differences between day and night collections for those species susceptible to drop traps (Sheridan et al., 1997). All water was pumped out of the drop trap through a 1-mm mesh plankton net into a removable mesh bag. Any nekton remaining on the bottom was removed and added to the mesh bag, and the contents were preserved in 10% formalin. For the purposes of this study, fishery species were defined as those organisms of recreational and commercial value monitored by Texas Parks and Wildlife Department or National Marine Fisheries Service (Campbell et al., 1991; National Marine Fisheries Service, 2000; Robinson et al., 1997). Fishery species captured in this study included brown shrimp *Farfantepenaeus aztecus*, pink shrimp *F. duorarum*, white shrimp *Litopenaeus setiferus* (following taxonomic revisions for penaeid shrimps by Farfante and Kensley, 1997), blue crab *Callinectes sapidus*, spotted seatrout *Cynoscion nebulosus*, spot *Leiostomus xanthurus*, Atlantic croaker *Micropogonias undulatus*, sheepshead *Archosargus probatocephalus*, southern flounder *Paralichthys lethostigma*, and gulf menhaden *Brevoortia patronus*.

2.3. Data analysis

One way analysis of variance (ANOVA) by sampling period was used to assess effects of habitat type on sediment and water column characteristics, seagrass coverage and biomass, and benthos and nekton densities. Data were transformed prior to ANOVA, using either arcsine square root for proportions or log $(x+1)$ for biomass and densities. Multiple comparison of treatment means employed Scheffé's test (Day and Quinn, 1989). Transformations were not always successful in effecting homogeneity of variance for faunal characteristics. Therefore a more conservative significance level was selected ($p \leq 0.025$) for benthos and nekton than for water, sediment, and seagrass ($p \leq 0.05$), as suggested by Underwood (1981), although this reduced the power of the tests. Similarity of faunal communities among sampling periods was compared using benthos or nekton densities pooled over all samples in each habitat type on a given date, unweighted pair-group average cluster analysis of Euclidean distances, and non-metric multi-dimensional scaling (MDS) of the resultant distance matrices. Stress values <0.2 were maintained in MDS to ensure valid representation of sample relationships (Clarke, 1993). Nonlinear regression was used to forecast seagrass coverage, shoot and root biomasses, and RSR as well as total densities of major taxa (annelids, benthic non-decapod crustaceans, molluscs, fishes, and decapods) in Maximum Impact habitat. January 1995 was assumed to be the starting date, and an asymptote of 100% equivalence to Natural Seagrass habitat was assumed to occur after

120 months as suggested by Rickner (1979). Schooling fishes (gulf menhaden and bay anchovy *Anchoa mitchilli*) were removed from this analysis because of one-time, high-density occurrences discussed below and because they are not usually associated with seagrass beds (Minello, 1999). Regression models were stated as:

$$y = b_0 - b_0/[1 + (x/b_2)^{b_1}]$$

where x = age in months, y = Maximum Impact value as a percent of Natural Seagrass value, b_0 = expected value at asymptote, b_2 = age at half-asymptote, and b_1 = slope. Regressions were used to predict when these variables reached 90% of Natural Seagrass values, indicating nearly complete recovery. Statistical analyses were conducted using Statistica (StatSoft, Inc., 1997). Tables and figures present non-transformed means.

3. Results

3.1. Dredged material

Qualitative surveys of the placement areas at the start of sampling in September 1995 indicated fresh deposits ranging in estimated size from 0.14 ha at 221 to 18.7 ha at 197. Non-vegetated areas generally trended downward over time for all sites except 197 (Fig. 2). Declines were relatively rapid at 211 and 221. The deposit at 211 was placed on the leeward side of islands created during the original dredging of the GIWW, so the dredged material was most likely compacted and re-vegetated by seagrasses rather than dispersed by winds and waves. The deposit at 221 was unprotected by any island and was exposed to a long fetch of southeasterly winds, which predominate this coastline (National Climatological Data Center, Local Climatological Data, available at http://www4.ncdc.noaa.gov). Sediments at this site were probably dispersed by winds and wave action. Declines in non-vegetated area at 187, 194, and 222 were relatively slow (Fig. 2). Deposits at 187 and 194 were placed around the northern, eastern, and southern edges of islands of emergent dredged material, thus they were relatively unprotected from southeasterly winds. Rapid re-vegetation was noted at these two sites between the final two sampling periods, presumably indicating some degree of sediment stability. Placement at 222 resulted in deposits on the southern and northern ends of two previously created dredged material islands separated by a narrow, shallow channel. The southern deposit appeared to be spreading over time as it washed off the end of the island, while the northern deposit re-vegetated rapidly, with the net effect of slow overall decrease in non-vegetated area. Again, the rapid re-vegetation noted at 222 between the final two sampling periods presumably indicated some degree of sediment stability. Changes in non-vegetated deposit area at 197 indicated the slowest re-vegetation over time (Fig. 2). This deposit had the largest estimated area, and it was placed on the windward side of an old dredged material island experiencing a long southeast fetch. The variation in estimated non-vegetated area between sampling periods indicated that materials may have been spreading out over time or that the single length × width measurements each month were insufficient to estimate area. Although non-vegetated area at 197 declined by 45% over time, the slow re-vegetation was most likely due to wind-induced sediment dispersal. The pooled effect across all six sites was a sharp drop in non-vegetated deposit area between September 1995 and April 1996 (60% of initial area), a relatively stable period of 18 months (due to measurements at 197), and a final sharp drop in non-vegetated area between September 1997 and April 1998 (50% of remaining area; Fig. 2). The end result was an apparent 75% reduction in non-vegetated deposit area after three years. This pattern should be viewed as preliminary, since areas were only estimated qualitatively.

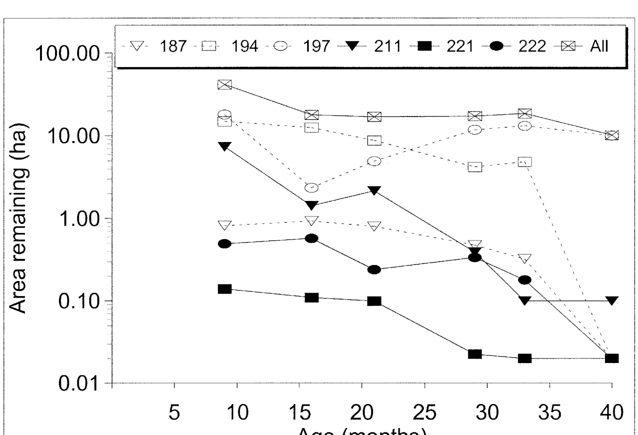

Fig. 2. Estimated non-vegetated area (ha) remaining at dredged material placement areas in Laguna Madre, assuming placement date of January 1995.

3.2. Surface sediments

Differences in sediment characteristics among habitat types were mostly detected early in the study (Fig. 3). Mean sand content ranged from 53% to 75%, and significant differences among habitat types were found during the first three sampling periods; however, those differences were not systematic. No significant differences were found in mean silt content. Mean clay content ranged between 16% and 36% and was significantly greater in the Maximum Impact habitat than elsewhere during the first two sampling periods. Differences in organic content were detected among habitat types in most sampling periods, with the only consistent

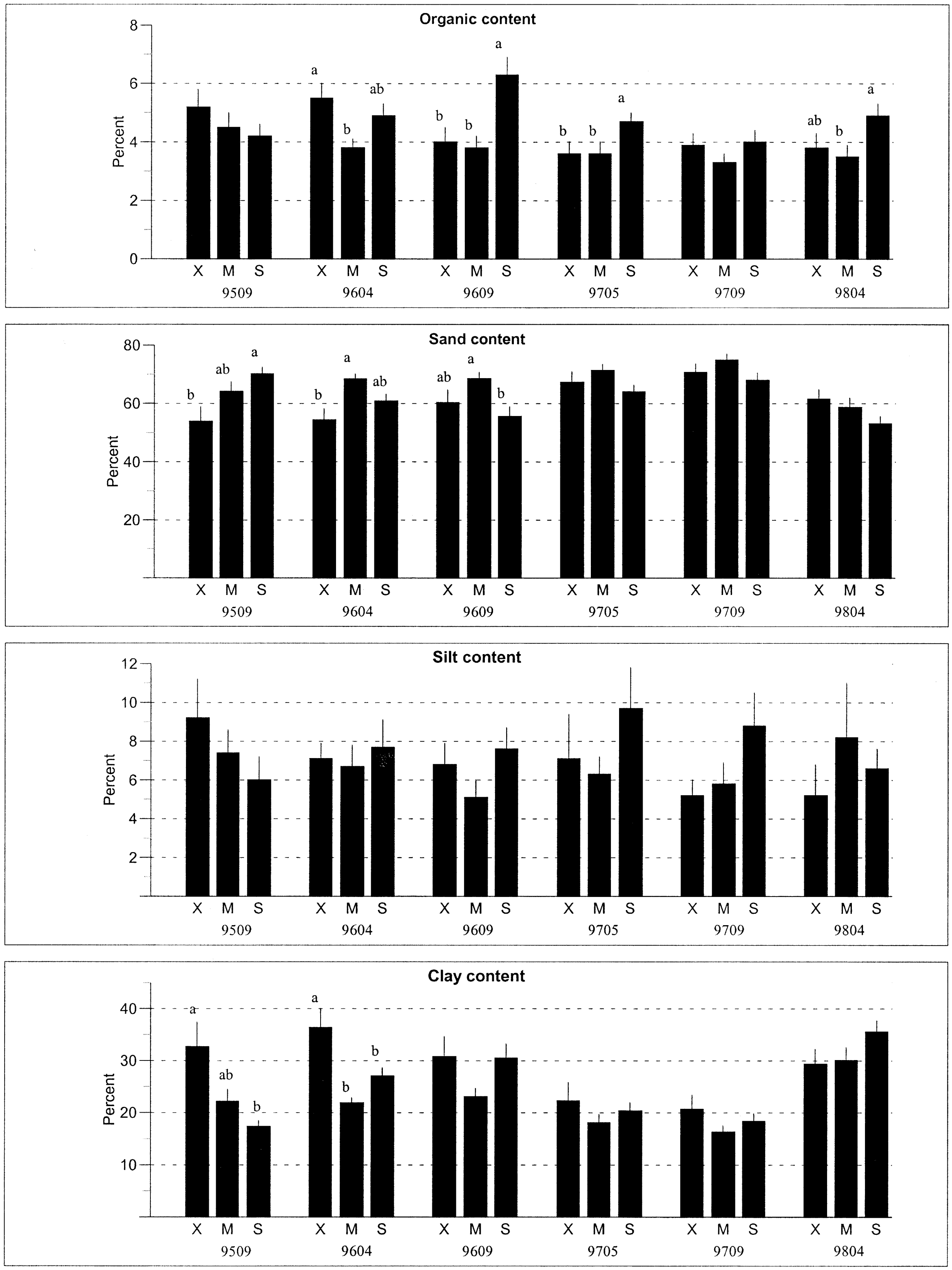

Fig. 3. Sediment characteristics (means + standard errors) of Maximum Impact (X), Minimum Impact (M), and Natural Seagrass (S) habitats during six sampling periods (YYMM) after dredged material placement. $N = 28–30$ per habitat per period. Means in each triplet with differing letters are significantly different ($p < 0.05$).

difference being that Natural Seagrass habitat had greater organic content than did Minimal Impact habitat.

3.3. Water column

Water column characteristics were similar among habitat types during each sampling period, with the exception of depth (Fig. 4). There were some significant differences in water temperature, but differences were not systematic and were only on the order of 1.0–1.5 °C. No differences in salinity were recorded. Turbidity was greater at Maximum Impact habitats but only during the first two sampling periods. Significant differences in depth were recorded, but these were primarily associated with Natural Seagrass habitats being deeper by 8–12 cm than dredged material deposits (as might be expected) and differences were not detected late in the study.

Brown tide bloomed at sites in Upper Laguna Madre during the preliminary survey in August 1995, was visible during the first four sampling periods, abated during the summer of 1997, and was not seen in the final two sampling periods. Brown tide was visible at Lower Laguna Madre sites only in May 1997. These algal blooms may have increased turbidity values; however, the blooms appeared to be in all habitat types when visible.

3.4. Seagrass beds

Seagrass coverage was relatively complete in both Natural Seagrass and Minimal Impact habitats, although coverage was as low as 90% in the latter habitat on two occasions (Fig. 5). Coverage in Maximum Impact habitats was significantly lower than elsewhere for the duration of the study. However, there were signs of steady re-colonization in the latter half of the study period, and mean coverage of Maximum Impact sites reached 48% approximately three years after dredging. Regression of seagrass cover in Maximum Impact habitat on months since dredging yielded the equation:

$$\text{Seagrass cover} = 100.2 - 100.2/[1 + (\text{Months})/40.8)^{5.6}],\quad R^2 = 0.993.$$

From this relationship, seagrass cover was predicted to reach 90% of Natural Seagrass at approximately 60 months post-dredging.

While coverage was relatively high in the Minimal Impact habitat, seagrass shoot and root biomasses were almost always significantly lower than in Natural Seagrass and were significantly lower still in the Maximum Impact habitat. Root:shoot ratios (RSR) remained significantly lower in Maximum Impact habitats than elsewhere throughout the study, although an increase was noted by the final sampling date (Fig. 5). Regression of seagrass shoot and root biomass and RSR in Maximum Impact habitat on months since dredging yielded the equations:

$$\text{Shoot biomass} = 104.6 - 104.6/[1 + (\text{Months})/62.9)^{4.7}],\quad R^2 = 0.998,$$

$$\text{Root biomass} = 99.9 - 99.9/[1 + (\text{Months})/44.4)^{9.3}],\quad R^2 = 0.999,\quad \text{and}$$

$$\text{RSR} = 100.5 - 100.5/[1 + (\text{Months})/37.2)^{4.6}],\quad R^2 = 0.959.$$

From these relationships, predicted times to reach 90% of Natural Seagrass were: shoot biomass = 93 months post-dredging, root biomass = 56 months, and RSR = 60 months.

Seagrasses began colonizing Maximum Impact habitats almost immediately. Clovergrass *Halophila engelmannii* found at 197 was the only seagrass noted in this habitat at any site during the first sampling period (September 1995). By April 1996, shoalgrass *Halodule wrightii* was found in Maximum Impact habitat at five of six sites. Shoalgrass was the predominant seagrass in the study areas and was found in all habitats and sites except the Maximum Impact habitat at 197, where clovergrass was the only colonist during the study period. Clovergrass was also found frequently in Minimum Impact habitats. Manatee grass *Syringodium filiforme* was found regularly in Minimum Impact and Natural Seagrass habitats in Lower Laguna Madre at 221 and 222 but only once in Upper Laguna Madre in Natural Seagrass habitat near 197. Turtlegrass *Thalassia testudinum* was recorded only in Natural Seagrass habitat near 221 and 222. Wigeongrass *Ruppia maritima*, a salt-tolerant freshwater plant, was recorded during the second half of the study when it was common in Minimum Impact habitat at all sites.

3.5. Benthic communities

A total of 220 taxa comprising 77,648 individuals was collected over the six sampling periods (a complete list of all taxa by habitat type is available from the author). Most benthic organisms were annelids (59% of the total), followed by non-decapod crustaceans (33%), molluscs (6%), and miscellaneous taxa (1%). Both Natural Seagrass and Minimal Impact habitats yielded about 2.5 times as many organisms as the Maximum Impact habitat over all collections. Approximately equal numbers of taxa were recorded from each habitat type over all samples: Maximum Impact, 177; Minimum Impact, 179; and Natural Seagrass, 160. Most organisms were relatively rare, as 11 taxa composed more than 70% of the total number collected.

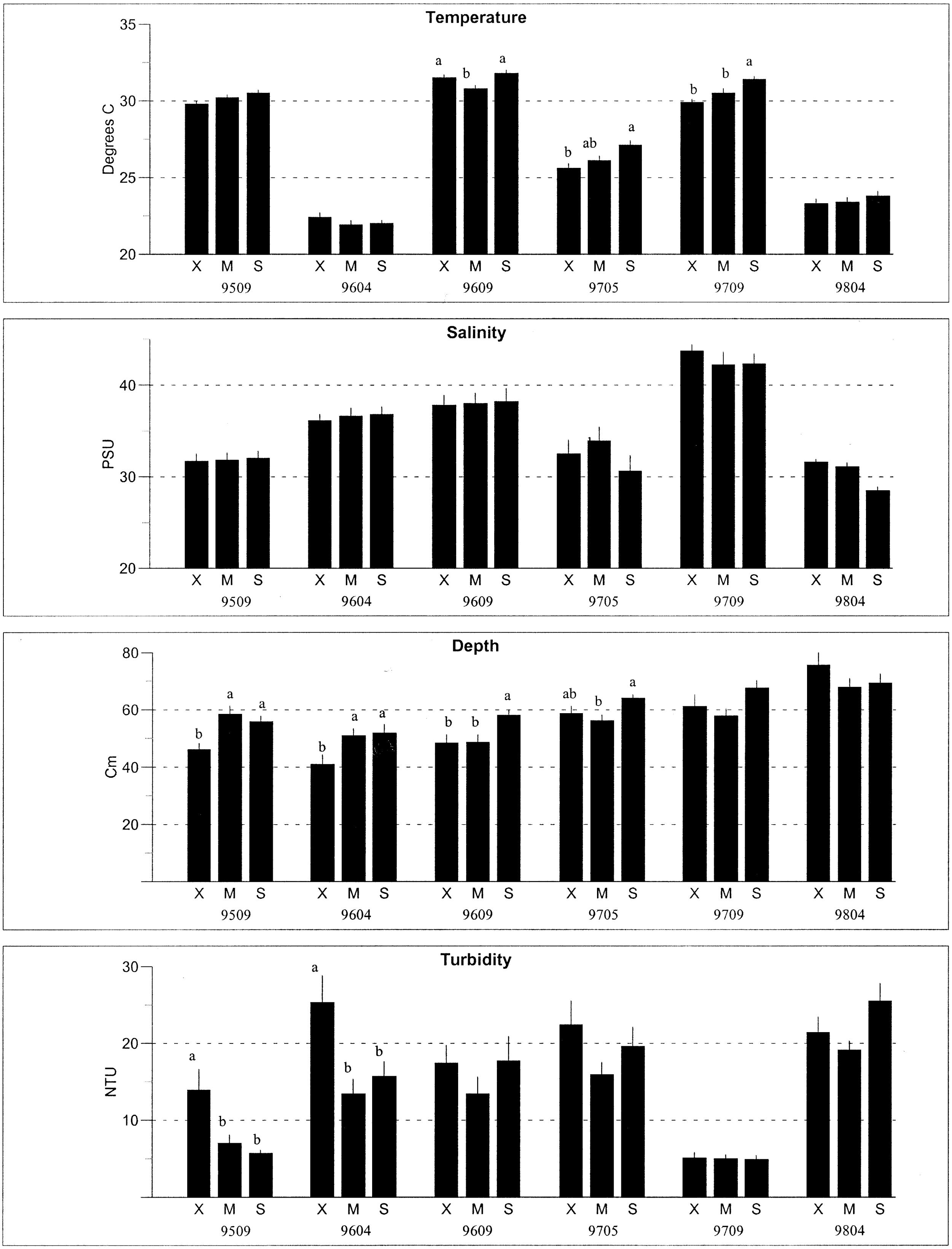

Fig. 4. Water column characteristics (means + standard errors) of Maximum Impact (X), Minimum Impact (M), and Natural Seagrass (S) habitats during 6 sampling periods (YYMM) after dredged material placement. $N = 29–30$ per habitat per period, except turbidity = 15 during April 1996. Means in each triplet with differing letters are significantly different ($p<0.05$).

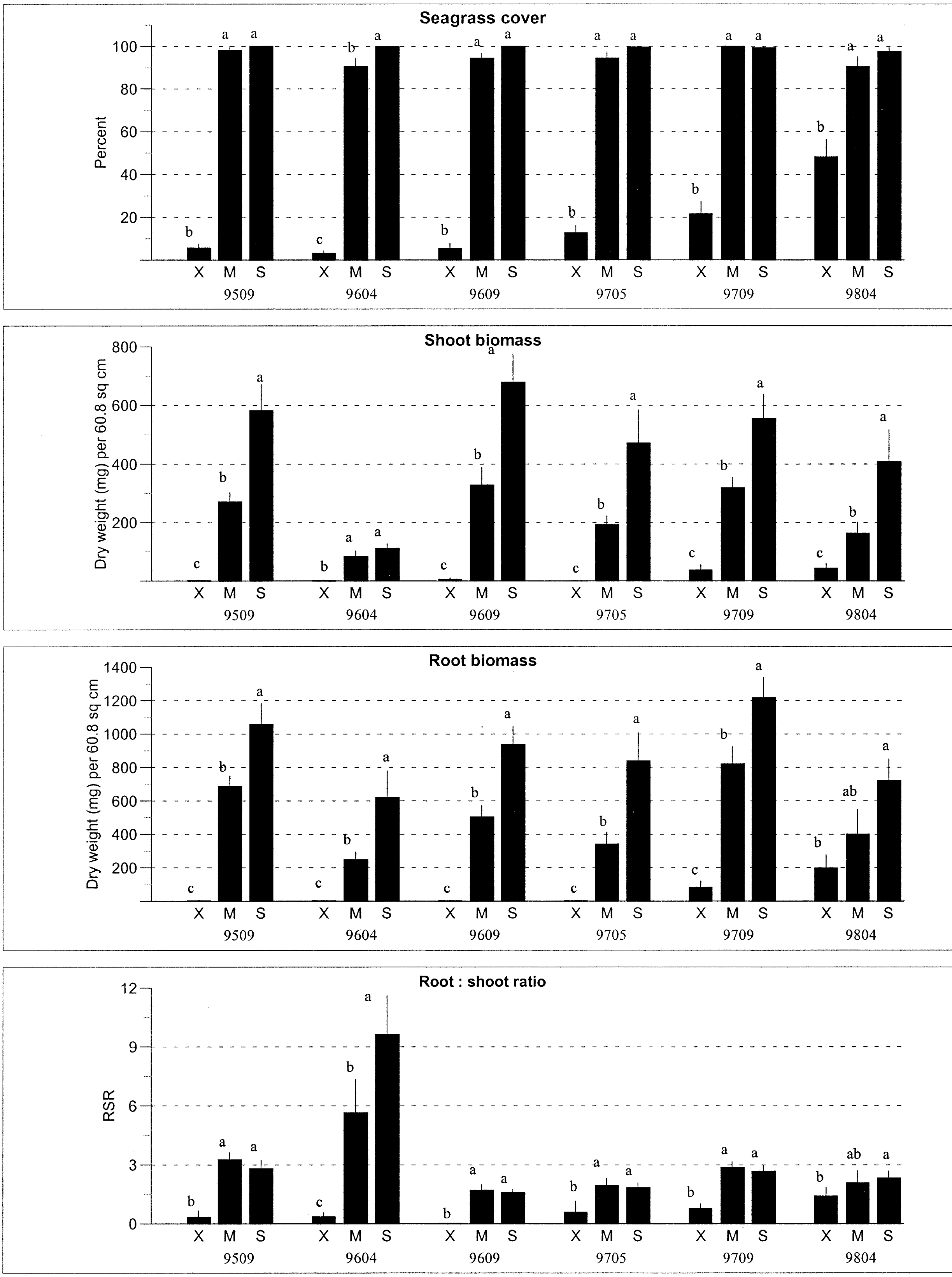

Fig. 5. Seagrass characteristics (means + standard errors) of Maximum Impact (X), Minimum Impact (M), and Natural Seagrass (S) habitats during six sampling periods (YYMM) after dredged material placement. $N = 29–30$ per habitat per period, except biomass = 20–21 for M and S during September 1997. Means in each triplet with differing letters are significantly different ($p < 0.05$).

Natural Seagrass habitat yielded three times as many annelids and Minimum Impact habitat yielded two times as many annelids as Maximum Impact habitat over all collections (21,975, 16,267, and 7702 individuals, respectively). Densities of total annelids were always significantly greater in Natural Seagrass habitat than in Maximum Impact habitat and were almost always so in Minimum Impact habitat (Table 1). Some of these differences in density were relatively large, for example by factors of 5–6 in May 1997 and September 1997. Regression of relative annelid densities in Maximum Impact habitat on months since dredging produced no significant model, because densities exhibited a sinusoidal pattern instead of an upward trend over time.

There were seven taxa of annelids among the 11 most abundant taxa found over all collections (Table 1). Mean densities of *Prionospio heterobranchia*, unidentified Oligochaeta, *Syllis cornutus*, and *Chone* cf. *americana* were usually significantly greater in Natural Seagrass or Minimum Impact habitats or both than in Maximum Impact habitat. Mean densities of *Capitella capitata*, *Exogone dispar*, and *Streblospio benedicti* were occasionally significantly greater in Natural Seagrass or Minimum Impact habitats than in Maximum Impact habitat.

Minimum Impact habitat yielded three times as many non-decapod crustaceans and Natural Seagrass habitat yielded two times as many crustaceans as Maximum Impact habitat over all samples (13,642, 8496, and 3804 individuals, respectively). Mean densities of total crustaceans were usually significantly greater in Minimum Impact habitat than in Maximum Impact habitat and were occasionally significantly greater in Natural Seagrass habitat (Table 1). Differences in mean density were relatively large in both September 1995 and April 1998, when large numbers of the amphipod *Cerapus benthophilus* were found inhabiting tubes attached to seagrass leaves. *Cerapus benthophilus* was often significantly more abundant in Minimum Impact habitat than elsewhere. Two other amphipods were also relatively abundant. Mean densities of *Grandidierella bonnieroides* were occasionally significantly greater in Natural Seagrass or Minimum Impact habitats than in Maximum Impact habitat. Mean densities of *Ampelisca* spp. (probably *A. abdita* but many were juveniles) were similar among habitat types during all sampling periods. Regression of relative crustacean densities in Maximum Impact habitat on months since dredging produced no significant model, because densities exhibited a declining pattern instead of an upward trend over time.

More molluscs were collected from Maximum Impact and Minimum Impact habitats than from Natural Seagrass (1613, 1676, and 1337 individuals, respectively). Mean densities of molluscs in general and of the dominant bivalve *Anomalocardia aubergiana* were significantly related to habitat only once or twice (Table 1). Regression of relative mollusc densities in Maximum Impact habitat on months since dredging produced no significant model, because densities generally equaled or exceeded those in Natural Seagrass.

Two groups of habitat-related benthic communities were identified through cluster analysis and multi-dimension scaling (MDS) of the pooled temporal data (Fig. 6). Benthos Group 1 encompassed closely related communities found on all the six collection dates from Maximum Impact habitats plus one Minimal Impact date. Benthos Group 2 was a loose aggregation of the remaining five Minimum Impact and all six Natural Seagrass communities. Chaining in cluster analysis and spatial spread of points in MDS indicated no obvious patterns in sampling dates for the latter group. Benthos Group 1 was characterized by relatively low densities of most dominant organisms mentioned previously plus organisms abundant in the other two habitat types such as *Grubeosyllis clavata* (annelid), Caprellidae (crustacean), *Amygdalum papyria* (mollusc), and Nemertinea. Overall densities of *Mediomastus ambiseta* and *Sabaco elongatus* (annelids) and *Corophium* spp. (crustaceans) were greater in Benthos Group 1 than in Benthos Group 2. Distinguishing characteristics of Benthos Group 2 (Fig. 6) included pronounced increases in densities of *Capitella capitata*, *Chone* cf. *americana*, *Heteromastus filiformis*, *Melinna maculata*, *Streblospio benedicti*, *Syllis cornutus*, and *Trypanosyllis vittigera* (annelids), *Ampelisca* spp., *Cerapus benthophilus*, *Elasmopus levis*, *Erichsonella attenuata*, *Grandidierella bonnieroides*, *Hargeria rapax*, *Harrieta faxoni*, and *Mucrogammarus mucronatus* (crustaceans), and *Mulinia lateralis* (mollusc) over those found in Benthos Group 1. In addition, Benthos Group 2 was characterized by one-time occurrences at relatively high densities (2–5 times greater than at other times) of the frequently occurring annelids *Naineris bicornis* (Natural Seagrass, May 1997), Oligochaeta (Natural Seagrass, September 1995), *Polydora cornuta* (Minimum Impact, September 1996), and *Polydora socialis* (Minimum Impact, September 1997), as well as the rarely occurring annelids *Hydroides dianthus* (Natural Seagrass, September 1996, 1404 of 1461 individuals), *Naineris dendritica* (Natural Seagrass, April 1998, 697 of 710 individuals), and *Ceratonereis irritabilis* (Minimal Impact, April 1998, 166 of 177 individuals) and the mollusc *Batillaria minima* (Minimal Impact, September 1995, 176 of 182 total individuals).

3.6. Nekton communities

Nekton sampling collected 47 fish taxa among 5353 individuals and 33 decapod taxa among 15,312 individuals (a complete list of all taxa by habitat type is available from the author). Both Natural Seagrass and Minimal Impact habitats yielded more than twice as many fishes as the Maximum Impact habitat (2032,

Table 1
Mean densities (number per 60.8 cm^2 $\pm$ 1 standard error) of annelids, non-decapod crustaceans, molluscs, and the 11 most abundant benthic taxa by sampling period and habitat (Max = Maximum Impact, Min = Minimum Impact, Nat = Natural Seagrass)[a]

Taxon	Habitat	Sep 1995		Apr 1996		Sep 1996		May 1997		Sep 1997		Apr 1998	
Annelids	Max	33.6 $\pm$ 5.9	b	61.9 $\pm$ 13.2	b	45.7 $\pm$ 7.7	b	15.0 $\pm$ 3.1	b	25.4 $\pm$ 6.0	b	74.8 $\pm$ 16.1	b
	Min	81.6 $\pm$ 9.1	a	88.9 $\pm$ 19.2	ab	97.7 $\pm$ 18.4	a	48.7 $\pm$ 6.6	a	132.2 $\pm$ 22.9	a	97.2 $\pm$ 19.9	b
	Nat	96.5 $\pm$ 12.4	a	113.8 $\pm$ 22.8	a	128.3 $\pm$ 24.9	a	117.5 $\pm$ 17.4	a	136.8 $\pm$ 16.2	a	147.3 $\pm$ 17.5	a
Prionospio heterobranchia	Max	3.6 $\pm$ 1.2	b	6.5 $\pm$ 2.2		5.0 $\pm$ 1.3	b	0.7 $\pm$ 0.4	b	5.8 $\pm$ 2.1	b	4.1 $\pm$ 1.4	b
	Min	15.3 $\pm$ 2.3	a	14.0 $\pm$ 4.4		35.2 $\pm$ 12.1	a	6.9 $\pm$ 2.8	a	35.7 $\pm$ 8.0	a	13.8 $\pm$ 4.2	a
	Nat	12.6 $\pm$ 2.3	a	16.1 $\pm$ 4.7		21.9 $\pm$ 6.4	ab	12.0 $\pm$ 3.5	a	35.3 $\pm$ 4.4	a	25.9 $\pm$ 6.1	a
Oligochaeta	Max	2.6 $\pm$ 1.0	b	3.9 $\pm$ 0.8	b	3.2 $\pm$ 0.8	b	1.7 $\pm$ 1.0	b	1.3 $\pm$ 0.5	b	1.7 $\pm$ 0.4	b
	Min	25.1 $\pm$ 5.7	a	23.1 $\pm$ 4.5	a	6.9 $\pm$ 1.5	ab	10.4 $\pm$ 3.3	a	13.5 $\pm$ 10.1	ab	3.1 $\pm$ 0.8	b
	Nat	40.1 $\pm$ 12.5	a	24.9 $\pm$ 7.8	a	23.1 $\pm$ 6.0	a	23.1 $\pm$ 8.6	a	21.2 $\pm$ 7.4	a	11.3 $\pm$ 3.3	a
Syllis cornutus	Max	0.5 $\pm$ 0.2	b	3.7 $\pm$ 1.4		2.0 $\pm$ 0.7	b	0.1 $\pm$ 0.1	b	0.8 $\pm$ 0.3	b	0	
	Min	4.9 $\pm$ 2.8	a	2.7 $\pm$ 0.8		10.2 $\pm$ 2.2	a	3.7 $\pm$ 1.0	a	16.0 $\pm$ 3.8	a	0	
	Nat	13.5 $\pm$ 3.0	a	8.3 $\pm$ 2.2		10.8 $\pm$ 1.7	a	4.4 $\pm$ 1.4	a	11.1 $\pm$ 2.5	a	0	
Chone cf. *americana*	Max	1.9 $\pm$ 0.6		5.7 $\pm$ 2.7		0.2 $\pm$ 0.1	b	0.8 $\pm$ 0.6	b	0.8 $\pm$ 0.4	b	0.9 $\pm$ 0.4	b
	Min	1.5 $\pm$ 0.3		5.5 $\pm$ 1.4		0.8 $\pm$ 0.3	ab	2.4 $\pm$ 0.8	a	2.0 $\pm$ 0.5	ab	5.1 $\pm$ 1.0	a
	Nat	1.7 $\pm$ 0.4		8.1 $\pm$ 1.9		2.4 $\pm$ 0.9	a	4.9 $\pm$ 1.6	a	7.3 $\pm$ 1.9	a	13.7 $\pm$ 4.5	a
Capitella capitata	Max	4.0 $\pm$ 1.0		6.5 $\pm$ 1.3		3.6 $\pm$ 0.6		1.5 $\pm$ 0.4	b	3.7 $\pm$ 1.3		2.2 $\pm$ 0.8	b
	Min	4.8 $\pm$ 1.6		6.7 $\pm$ 1.7		4.3 $\pm$ 1.3		3.0 $\pm$ 0.8	ab	4.9 $\pm$ 1.4		3.6 $\pm$ 1.3	ab
	Nat	7.0 $\pm$ 1.5		4.0 $\pm$ 0.6		2.8 $\pm$ 0.6		14.1 $\pm$ 3.4	a	10.9 $\pm$ 3.2		4.6 $\pm$ 1.0	a
Exogone dispar	Max	1.7 $\pm$ 1.1	b	0.4 $\pm$ 0.2	b	0		0.2 $\pm$ 0.1		2.5 $\pm$ 1.7	b	2.0 $\pm$ 0.6	b
	Min	10.3 $\pm$ 2.2	a	7.4 $\pm$ 2.5	a	0.3 $\pm$ 0.2		0.8 $\pm$ 0.4		14.1 $\pm$ 4.3	a	3.0 $\pm$ 0.6	ab
	Nat	2.0 $\pm$ 0.5	b	3.9 $\pm$ 1.0	ab	0.3 $\pm$ 0.1		2.5 $\pm$ 1.4		13.1 $\pm$ 5.7	a	3.6 $\pm$ 0.7	a
Streblospio benedicti	Max	12.1 $\pm$ 3.9		11.9 $\pm$ 4.3	b	12.5 $\pm$ 2.9		0.5 $\pm$ 0.2	b	2.0 $\pm$ 0.9		20.2 $\pm$ 11.4	
	Min	10.4 $\pm$ 3.5		15.9 $\pm$ 14.2	b	11.4 $\pm$ 3.8		2.7 $\pm$ 1.3	ab	6.6 $\pm$ 5.4		21.3 $\pm$ 9.1	
	Nat	13.2 $\pm$ 3.5		39.0 $\pm$ 18.0	a	15.8 $\pm$ 5.9		16.5 $\pm$ 5.3	a	15.7 $\pm$ 4.9		30.3 $\pm$ 12.1	
Crustaceans	Max	19.3 $\pm$ 5.1	b	61.1 $\pm$ 18.2		10.3 $\pm$ 2.6		8.3 $\pm$ 2.9	b	3.1 $\pm$ 0.8	b	25.2 $\pm$ 8.0	b
	Min	165.8 $\pm$ 80.5	a	66.8 $\pm$ 14.4		26.3 $\pm$ 5.2		28.7 $\pm$ 6.6	a	25.8 $\pm$ 5.0	a	143.7 $\pm$ 34.6	a
	Nat	21.4 $\pm$ 3.6	b	66.3 $\pm$ 12.7		9.1 $\pm$ 1.7		15.9 $\pm$ 3.2	ab	34.9 $\pm$ 11.9	a	138.9 $\pm$ 52.9	a
Cerapus benthophilus	Max	2.7 $\pm$ 1.4	b	6.8 $\pm$ 3.0	b	0.2 $\pm$ 0.2	b	0.6 $\pm$ 0.4	b	0	b	17.1 $\pm$ 7.3	b
	Min	147.4 $\pm$ 79.9	a	22.9 $\pm$ 8.9	a	6.5 $\pm$ 3.6	a	12.2 $\pm$ 6.0	a	2.9 $\pm$ 1.7	ab	100.0 $\pm$ 32.3	a
	Nat	4.3 $\pm$ 1.2	b	8.4 $\pm$ 2.9	b	0.1 $\pm$ 0.1	b	0.3 $\pm$ 0.2	b	11.7 $\pm$ 9.3	a	111.8 $\pm$ 51.7	a
Grandidierella bonnieroides	Max	4.6 $\pm$ 1.4		4.5 $\pm$ 1.7	b	3.6 $\pm$ 1.0	b	0.7 $\pm$ 0.2		0.7 $\pm$ 0.3	b	0.9 $\pm$ 0.3	
	Min	4.7 $\pm$ 1.0		7.7 $\pm$ 3.3	a	10.0 $\pm$ 1.7	a	2.4 $\pm$ 0.6		4.5 $\pm$ 1.1	a	1.3 $\pm$ 0.4	
	Nat	4.9 $\pm$ 1.8		5.7 $\pm$ 1.1	ab	4.6 $\pm$ 0.7	b	5.5 $\pm$ 2.6		5.6 $\pm$ 1.4	a	1.2 $\pm$ 0.4	
Ampelisca spp.	Max	8.9 $\pm$ 3.1		42.1 $\pm$ 12.6		1.0 $\pm$ 0.3		4.6 $\pm$ 2.6		0		3.3 $\pm$ 1.1	
	Min	1.5 $\pm$ 0.3		26.4 $\pm$ 10.6		2.4 $\pm$ 1.0		8.4 $\pm$ 2.7		0		29.1 $\pm$ 14.3	
	Nat	0.7 $\pm$ 0.2		37.2 $\pm$ 10.6		0.9 $\pm$ 0.2		1.8 $\pm$ 0.5		0		12.9 $\pm$ 4.3	
Molluscs	Max	3.8 $\pm$ 0.8		11.9 $\pm$ 3.3		4.6 $\pm$ 0.7	a	15.7 $\pm$ 4.1		8.5 $\pm$ 1.5		9.4 $\pm$ 1.8	
	Min	8.5 $\pm$ 5.8		8.7 $\pm$ 1.4		4.1 $\pm$ 0.6	a	11.3 $\pm$ 2.6		14.6 $\pm$ 2.9		9.6 $\pm$ 1.8	
	Nat	2.0 $\pm$ 0.5		11.3 $\pm$ 2.7		3.0 $\pm$ 1.1	b	9.3 $\pm$ 2.2		9.4 $\pm$ 2.6		10.5 $\pm$ 2.1	
Anomalocardia aubergiana	Max	1.4 $\pm$ 0.4	a	3.8 $\pm$ 1.0		3.6 $\pm$ 0.6	a	11.6 $\pm$ 3.6		6.5 $\pm$ 1.5		2.8 $\pm$ 0.6	
	Min	0.5 $\pm$ 0.2	b	2.5 $\pm$ 0.7		2.4 $\pm$ 0.4	ab	7.4 $\pm$ 2.2		5.8 $\pm$ 2.0		2.5 $\pm$ 0.6	
	Nat	0.9 $\pm$ 0.3	ab	1.4 $\pm$ 0.5		2.0 $\pm$ 0.9	b	6.6 $\pm$ 2.1		5.0 $\pm$ 1.8		4.3 $\pm$ 1.4	

[a] $N = 30$ per habitat per month, except $n = 29$ for May 1997 Min and September 1997 Min and $n = 27$ for September 1997 Nat. Means indicated with differing symbols in a given period are significantly different ($p \leq 0.025$).

2397, and 924 fishes, respectively) and about nine times as many decapods (6815, 7697, and 800 decapods, respectively). With one exception, mean fish and decapod densities were significantly lower in Maximum Impact habitat than elsewhere during each sampling period (Table 2). Habitat-related differences in fish density tended to be greater during the early sampling periods and narrower later in the study. Fish densities in the Maximum Impact habitat may have begun increasing by April 1998 as seagrass coverage increased (Table 2). Habitat-related differences in decapod density tended to be larger in fall (September) than in spring (April or May; Table 2). There was also an upward trend of decapod densities in Maximum Impact habitats during April 1998 that was likely connected to increasing seagrass coverage.

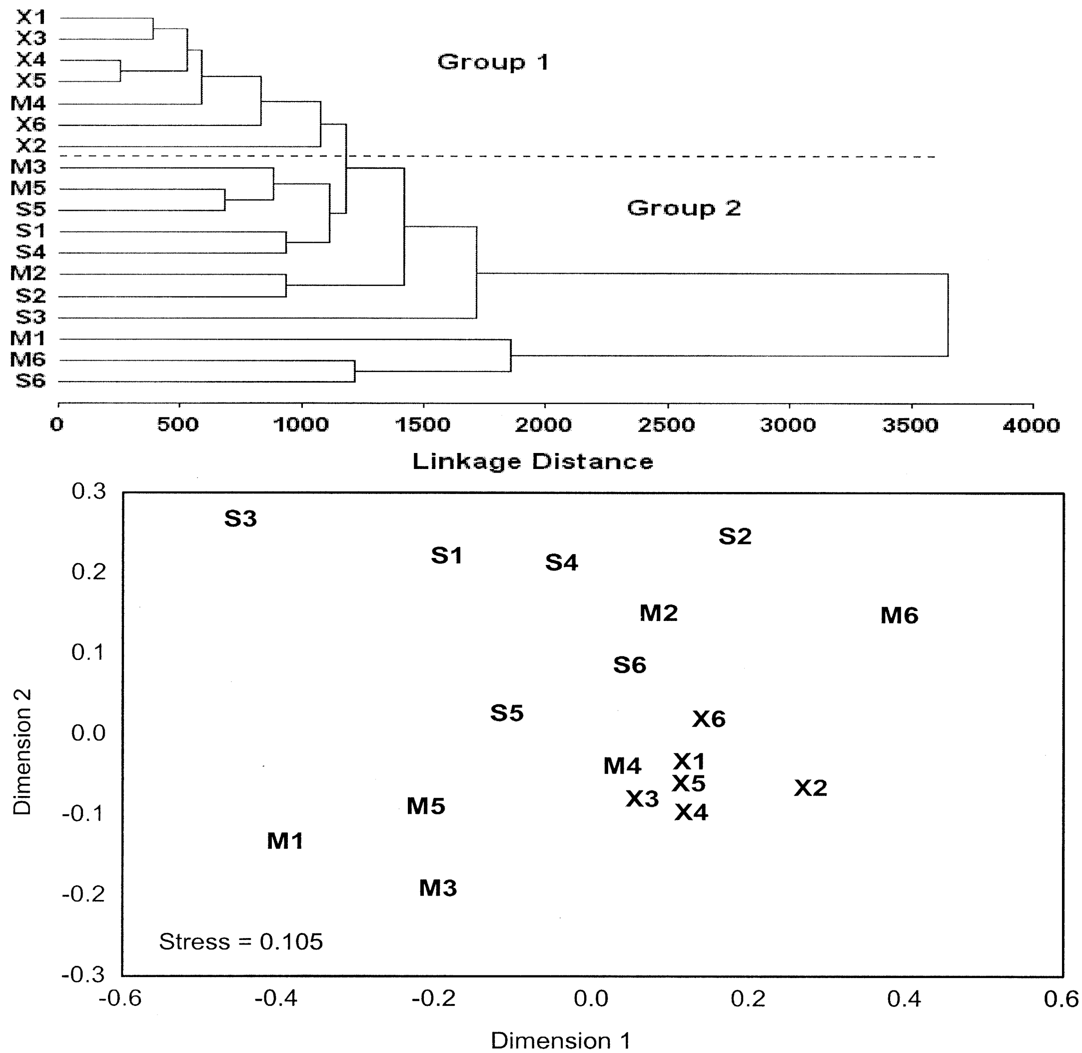

Fig. 6. Cluster analysis (upper) and MDS ordination (lower) of benthic communities by habitat type and sampling date using UPGMA linkage of a Euclidean distance matrix. X = Maximum Impact, M = Minimum Impact, S = Natural Seagrass. Sampling dates sequentially numbered (1 September 1995 to 6 April 1998).

Five fish taxa and nine decapod taxa comprised 91% of all nekton collected (Table 2). Bay anchovy *Anchoa mitchilli* and gulf menhaden *Brevoortia patronus* (both schooling species) were occasionally collected at high densities but there were no significant habitat-related differences. Mean densities of code goby *Gobiosoma robustum*, pinfish *Lagodon rhomboides*, and gulf pipefish *Syngnathus scovelli* were significantly greater in Natural Seagrass and Minimum Impact habitats than in Maximum Impact habitat during most sampling periods (Table 2). When abundant, mean densities of eight of nine decapod taxa were significantly greater in Natural Seagrass and Minimum Impact habitats than in Maximum Impact habitat (Table 2). These taxa included bigclaw snapping shrimp *Alpheus heterochaelis*, brown shrimp *Farfantepenaeus aztecus*, pink shrimp *Farfantepenaeus duorarum*, zostera shrimp *Hippolyte zostericola*, brackish grass shrimp *Palaemonetes intermedius*, daggerblade grass shrimp *Palaemonetes pugio*, blue crab *Callinectes sapidus*, and gulf grassflat crab *Dyspanopeus texana*. Densities of bigclaw snapping shrimp were likely underestimated because this is a burrowing species. Densities of arrow shrimp *Tozeuma carolinense* were usually significantly greater in Minimum Impact habitat than elsewhere. High densities of zostera shrimp and gulf grassflat crab were primarily responsible for the overall increase in decapod densities noted in Maximum Impact habitat during April 1998.

Two groups of nekton communities were identified through cluster analysis and MDS of the pooled temporal data (Fig. 7). Nekton Group 1 encompassed communities found on all six collection dates from Maximum Impact habitats plus two Minimal Impact dates. Nekton Group 2 included communities in Natural Seagrass during all six periods and in Minimum Impact habitats on four dates. Nekton Group 1 was

Table 2
Mean densities (number per $m^2 \pm 1$ standard error) of total fishes, total decapods, and the 14 most abundant taxa by sampling period and habitat (Max = Maximum Impact, Min = Minimum Impact, Nat = Natural Seagrass)[a]

Taxon	Habitat	Sep 1995		Apr 1996		Sep 1996		May 1997		Sep 1997		Apr 1998	
Fishes	Max	3.1 ± 1.0	b	1.3 ± 0.4	b	1.1 ± 0.4	b	17.8 ± 10.0		2.6 ± 0.8	b	4.8 ± 1.1	b
	Min	17.3 ± 4.7	a	18.0 ± 7.0	a	14.1 ± 3.6	a	8.9 ± 2.8		13.1 ± 1.6	a	8.5 ± 1.4	a
	Nat	11.2 ± 1.7	a	18.9 ± 3.4	a	12.1 ± 1.9	a	7.1 ± 0.8		8.8 ± 1.0	a	9.6 ± 1.2	a
Anchoa mitchilli	Max	2.1 ± 0.9		0.1 ± 0.1		0.4 ± 0.3		0		0		0.5 ± 0.2	
	Min	7.5 ± 4.3		0.3 ± 0.2		1.4 ± 0.9		0.2 ± 0.1		0		0.5 ± 0.3	
	Nat	2.3 ± 1.0		0		0.9 ± 0.5		0.6 ± 0.3		0		0.2 ± 0.1	
Brevoortia patronus	Max	0		0.6 ± 0.3		0		16.5 ± 10.0		0		1.2 ± 0.7	
	Min	0		4.1 ± 2.5		0		3.7 ± 2.6		0		0.3 ± 0.1	
	Nat	0		1.1 ± 0.9		0.1 ± 0.1		0.3 ± 0.2		0		0.1 ± 0.1	
Gobiosoma robustum	Max	0.2 ± 0.1	b	0		0.2 ± 0.1	b	0	b	1.9 ± 0.7	b	2.1 ± 0.9	b
	Min	3.8 ± 0.7	a	0.5 ± 0.2		9.2 ± 3.1	a	1.5 ± 0.4	a	9.5 ± 1.5	a	3.9 ± 0.8	a
	Nat	4.4 ± 1.1	a	0.3 ± 0.1		6.3 ± 1.5	a	1.8 ± 0.4	a	5.8 ± 0.9	a	4.9 ± 0.8	a
Lagodon rhomboides	Max	0		0.1 ± 0.1	b	0		0.4 ± 0.2	b	0.1 ± 0.1		0.3 ± 0.1	b
	Min	0.2 ± 0.1		11.3 ± 6.3	a	0.3 ± 0.1		1.4 ± 0.3	a	0.4 ± 0.1		0.4 ± 0.1	b
	Nat	0.1 ± 0.1		15.4 ± 3.4	a	0.6 ± 0.2		1.4 ± 0.4	a	0.2 ± 0.1		2.7 ± 0.8	a
Syngnathus scovelli	Max	0.1 ± 0.1	b	0		0.2 ± 0.1	b	0	b	0.1 ± 0.1	b	0.2 ± 0.1	b
	Min	3.6 ± 0.5	a	0.4 ± 0.1		1.4 ± 0.4	a	1.0 ± 0.3	a	1.7 ± 0.4	a	1.2 ± 0.2	a
	Nat	2.2 ± 0.4	a	0.6 ± 0.2		1.6 ± 0.2	a	1.4 ± 0.4	a	1.1 ± 0.2	a	1.1 ± 0.3	a
Decapods	Max	2.4 ± 0.6	b	2.7 ± 0.6	b	1.9 ± 0.8	b	1.3 ± 0.4	b	3.3 ± 1.0	b	15.1 ± 5.8	b
	Min	53.3 ± 5.7	a	39.0 ± 7.7	a	61.4 ± 17.9	a	19.5 ± 3.2	a	57.5 ± 11.1	a	25.0 ± 5.0	a
	Nat	57.4 ± 8.8	a	33.2 ± 2.6	a	43.7 ± 6.0	a	27.8 ± 3.8	a	33.0 ± 4.1	a	32.1 ± 5.5	a
Alpheus heterochaelis	Max	0	b	0		0	b	0		0.1 ± 0.1	b	0	
	Min	2.8 ± 0.7	a	0.1 ± 0.1		1.3 ± 0.5	a	0		3.0 ± 0.8	a	0.1 ± 0.1	
	Nat	1.5 ± 0.4	a	0.2 ± 0.1		1.4 ± 0.4	a	0.1 ± 0.1		0.8 ± 0.3	ab	0.3 ± 0.2	
Farfantepenaeus aztecus	Max	0.2 ± 0.1	b	0.5 ± 0.2	b	0.1 ± 0.1		0.4 ± 0.1	b	0.3 ± 0.2	b	1.1 ± 0.4	
	Min	1.4 ± 0.3	a	2.9 ± 0.5	a	0.4 ± 0.1		3.1 ± 0.7	a	1.2 ± 0.2	a	1.6 ± 0.5	
	Nat	1.0 ± 0.2	a	4.6 ± 0.8	a	0.8 ± 0.2		2.3 ± 0.5	a	0.9 ± 0.3	ab	1.2 ± 0.4	
Farfantepenaeus duorarum	Max	0.3 ± 0.2	b	0.5 ± 0.1	b	0	b	0.1 ± 0.1	b	0.4 ± 0.3		0.4 ± 0.2	
	Min	2.2 ± 0.5	a	1.7 ± 0.4	a	1.3 ± 0.4	a	1.1 ± 0.4	a	1.2 ± 0.4		1.4 ± 0.4	
	Nat	2.2 ± 0.6	a	2.2 ± 0.5	a	1.2 ± 0.3	a	0.9 ± 0.3	ab	0.9 ± 0.2		1.3 ± 0.3	
Hippolyte zostericola	Max	0.1 ± 0.1	b	0	b	0	b	0		0.2 ± 0.2	b	4.7 ± 2.3	b
	Min	11.6 ± 4.0	a	3.4 ± 1.2	a	8.0 ± 3.1	a	0.7 ± 0.3		17.6 ± 5.2	a	11.2 ± 2.8	a
	Nat	8.8 ± 3.4	a	2.1 ± 0.5	a	5.3 ± 1.9	a	0		10.4 ± 2.7	a	6.7 ± 1.8	a
Palaemonetes intermedius	Max	0	b	0	b	0.9 ± 0.6	b	0	b	0.6 ± 0.4	b	0.6 ± 0.3	b
	Min	16.9 ± 2.8	a	19.4 ± 5.3	a	18.7 ± 6.4	a	8.7 ± 2.1	a	17.1 ± 3.5	a	2.5 ± 0.9	b
	Nat	16.1 ± 2.8	a	16.1 ± 3.2	a	19.7 ± 2.4	a	18.5 ± 3.8	a	14.4 ± 2.1	a	10.6 ± 2.7	a
Palaemonetes pugio	Max	0.1 ± 0.1	b	0		0	b	0		0	b	0	b
	Min	2.9 ± 0.7	a	0.8 ± 0.2		3.4 ± 1.5	a	0.2 ± 0.1		1.7 ± 0.8	a	0.1 ± 0.1	b
	Nat	10.4 ± 2.8	a	0.7 ± 0.2		3.0 ± 1.2	a	0.5 ± 0.2		1.0 ± 0.4	a	1.2 ± 0.4	a
Tozeuma carolinense	Max	0.3 ± 0.2	b	0	b	0	b	0	b	0	b	0	
	Min	8.1 ± 3.1	a	1.9 ± 0.8	a	4.3 ± 2.2	a	2.1 ± 1.2	a	2.2 ± 1.1	a	0.7 ± 0.4	
	Nat	9.3 ± 4.0	a	0.4 ± 0.2	b	1.1 ± 0.4	b	0.1 ± 0.2	b	0.2 ± 0.1	b	0.4 ± 0.2	
Callinectes sapidus	Max	0.2 ± 0.1	b	0.3 ± 0.1	b	0.2 ± 0.1		0		0.3 ± 0.2		0.8 ± 0.4	b
	Min	2.1 ± 0.5	a	2.7 ± 0.8	a	1.2 ± 0.5		0.6 ± 0.2		0.5 ± 0.2		1.4 ± 0.5	a
	Nat	0.8 ± 0.3	b	2.1 ± 0.5	a	1.0 ± 0.4		0.3 ± 0.1		0.4 ± 0.2		1.7 ± 0.4	a
Dyspanopeus texanus	Max	0.3 ± 0.1	b	0	b	0.3 ± 0.3	b	0	b	0.5 ± 0.2	b	6.4 ± 3.3	
	Min	3.5 ± 0.8	a	3.1 ± 1.8	a	6.3 ± 2.7	a	1.7 ± 0.7	a	11.2 ± 5.8	a	3.7 ± 1.5	
	Nat	5.9 ± 1.3	a	2.7 ± 0.6	a	6.1 ± 3.3	a	3.9 ± 1.4	a	3.0 ± 0.7	a	3.2 ± 0.7	

[a] $N = 30$ per habitat per month. Means indicated with differing letters in a given period are significantly different ($p \leq 0.025$; tests not conducted if all means < 1 per m^2).

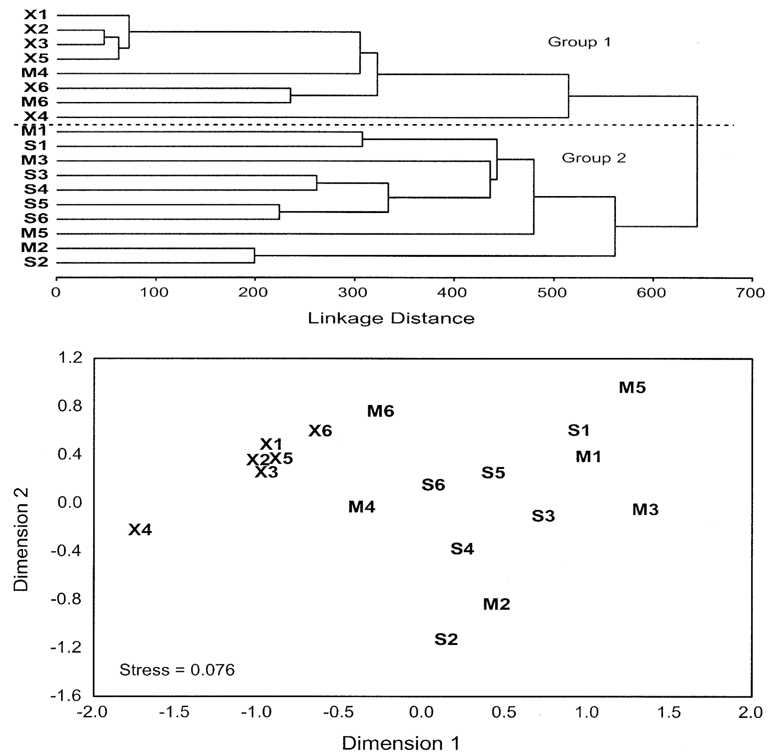

Fig. 7. Cluster analysis (upper) and MDS ordination (lower) of nekton communities by habitat type and sampling date using UPGMA linkage of a Euclidean distance matrix. X = Maximum Impact, M = Minimum Impact, S = Natural Seagrass. Sampling dates sequentially numbered (1 September 1995 to 6 April 1998).

characterized by relatively low densities of most dominant organisms mentioned previously plus other fishes numerous in the other two habitat types such as darter goby *Gobionellus boleosoma* and rainwater killifish *Lucania parva*. Densities of gulf menhaden were greater in Nekton Group 1 than in Nekton Group 2, but there was a one-time high catch of 59% of all gulf menhaden captured during May 1997 in Maximum Impact habitat. In addition, Nekton Group 2 was characterized by one-time occurrences at relatively high densities (2–5 times greater than at other times) of pinfish (Minimum Impact and Natural Seagrass, April 1996, 340 and 461 of 1057 total individuals), hermit crab *Pagurus criniticornis* (Minimum Impact, September 1996, 280 of 294 individuals), and ridgeback mud crab *Panopeus turgidus* (Minimum Impact, September 1996, 252 of 353 individuals).

Juveniles of a variety of fishery species were collected (Table 3). Gulf menhaden, blue crab, brown shrimp, and pink shrimp were collected at relatively high densities and were mentioned previously. All other taxa were characterized by relatively low densities and habitat-related differences were not significant. Sheepshead, spotted seatrout, and white shrimp were more numerous in Natural Seagrass or Minimum Impact habitats than in the Maximum Impact habitat. Spot were collected from all habitats, and Atlantic croaker and southern flounder were more numerous in Maximum Impact habitat.

Regression of relative fish and decapod densities in Maximum Impact habitat on months since dredging yielded the equations:

$$\text{Fish density} = 102.4 - 102.4/[1 + (\text{Months})/41.6)^{3.6}],$$
$$R^2 = 0.978, \text{ and}$$

$$\text{Decapod density} = 99.9 - 99.9/[1 + (\text{Months})/40.5)^{10.2}],$$
$$R^2 = 0.987.$$

From these relationships, fish and decapod densities were predicted to reach 90% of Natural Seagrass

Table 3
Total abundance of fishery species by habitat type. $N = 180$ per habitat

Taxon	Maximum Impact	Minimum Impact	Natural Seagrass
Archosargus probatocephalus	0	2	2
Brevoortia patronus	546	243	49
Cynoscion nebulosus	3	20	12
Leiostomus xanthurus	21	16	26
Micropogonias undulatus	17	3	7
Paralichthys lethostigma	5	2	1
Farfantepenaeus aztecus	78	319	322
Farfantepenaeus duorarum	54	268	264
Litopenaeus setiferus	9	17	17
Callinectes sapidus	57	254	189

densities at approximately 70 and 50 months post-dredging, respectively.

4. Discussion

The variety of system components studied here indicated that structural recovery from dredged material placement may have been nearly complete for some components after 1.5 years, whereas recovery for others extended beyond the three year study period. Total area of non-vegetated dredged material decreased by 75% over the study period (3+ years after deposition). Most of the remaining dredged material at five of the six sites was either in shallow or deep water not suitable for seagrasses. Re-vegetation at depth extremes might have been limited by tidal exposure or high water temperature at the shallow end or by reduced water clarity and light availability at depths >1 m if disturbances such as brown tide occurred. Total reduction in area of non-vegetated deposits at five sites was 86–99% over the study period. However, there was still a large non-vegetated deposit at one placement area (55% of estimated original area) 3+ years after deposition.

Water column turbidity and surface sediment characteristics in Maximum Impact habitats were similar to those in the surrounding Minimum Impact and Natural Seagrass habitats after 1.5 years. Prior to stabilization, sediments and water clarity are subject to wind-induced resuspension both from strong but short-lived northerly winds and from persistent southeasterly winds (Onuf, 1994; Brown and Kraus, 1996).

Seagrass colonization of Maximum Impact habitat was noticeable after two years of observation and was widespread after three years. This is the same time frame observed by Odum (1963) in an adjacent bay, and the rapidity of coverage at some sites between observation periods was similar to rapid growth onto dredged material observed by Circé (1979) over a two month period. The lag between dredging and seagrass coverage may be due to a combination of time of year, physical disturbance by winds and currents, and chemical inhibition by pore water constituents such as ammonium or sulfide, both of which can be toxic to seagrasses (Carlson et al., 1994, Van Katwijk et al., 1997). A study of dredged material placement sites in Lower Laguna Madre indicated high levels of ammonium (>900 μM) in recently placed sediments and elevated ammonium concentrations up to three years after placement (Sheridan and Minello, 2003). High sulfide concentrations also have been detected at placement areas (235–1125 μM H_2S; Dunton et al., 2003) that exceed those typical of Laguna Madre seagrass beds (<200 μM H_2S; Pulich, 1985). As these pore water constituents are released or oxidized, sediments presumably become more amenable for seagrass colonization. This would explain why seagrass shoot biomass, root biomass, and root:shoot ratio were increasing but remained significantly lower in Minimum Impact and Maximum Impact habitats than in Natural Seagrass even after three years.

Once seagrasses begin to cover the dredged material and form Minimum Impact habitat, increases in densities of the mobile macrofauna can be expected (Fonseca et al., 1990; Sheridan and Minello, 2003). Nekton communities in Minimum Impact and Natural Seagrass habitats generally were similar in density and species composition over the study period. Total fish and total decapod densities, as well as several of the more abundant species, remained significantly lower in the Maximum Impact habitat throughout the study but were showing signs of recovery after three years. The relatively sparse seagrasses growing in Maximum Impact habitats were likely beginning to provide structure and shelter for nekton.

However, one of the other functions ascribed to seagrass beds that is linked to their support of high nekton densities is the provision of food such as benthic organisms. Given the patterns in densities of major benthic groups and individual taxa, there is little indication that benthic communities in Maximum Impact habitats had begun to recover even after three years. The upturns in total benthos, annelid, and crustacean densities observed in April 1998 were similar to those observed in April 1996 and likely represented the remnants of the usual late winter–early spring peak in benthic faunal densities observed along the Texas coast (Harper, 1992). The May 1997 collections did not demonstrate this peak and may have occurred too late in the annual cycle. Mollusc densities were the exception, since there seemed to be no consistent relation between density and dredging impact in periodic collections and since total numbers collected were highest in impacted habitats. In addition, the benthic communities characterizing Maximum Impact habitats remained distinctive from other habitats over the sampling period.

Recovery of seagrass coverage and nekton densities in placement areas to 90% of those in undisturbed seagrasses was predicted to take 50–70 months (4.2–5.8 years). Recovery of densities of benthic annelids and non-decapod crustaceans could not be predicted, but it likely would take at least five years and perhaps as long as 10–20 years as proposed by Rickner (1979). It must be noted that Rickner's study had no placement areas between the ages of three months and 10 years. Thus, the impact to fishery and forage organisms declines over time, seemingly in tandem with re-vegetation of dredged material deposits.

Brown tide can affect the distribution and biomass of seagrasses. During brown tide events of 1990–1993, Dunton (1994) recorded a 50% decline in below-ground biomass of shoalgrass that was reflected in reduced root:shoot ratios. Continued brown tide resulted in mortality of over 9.4 km^2 of seagrass in waters deeper than 1.4 m by 1995 (Onuf, 1996). However, relatively rapid coverage of dredged materials by seagrasses at sites in Upper Laguna Madre was noted in this study, indicating that light reduction due to brown tide was not severe enough to prevent seagrass colonization of shallow (<1 m) dredged material deposits. Brown tide may also impact the Laguna Madre fauna, although direct toxic effects have not been found and food web disruptions are more likely (reviewed by Street et al., 1997). In the present study, there were no indications that any of the dominant benthos or nekton had been excluded from Upper Laguna Madre since these taxa were abundant in both systems.

The present dredging program for the Gulf Intracoastal Waterway in Laguna Madre is a five year cycle, in which high maintenance reaches may be dredged every two years and low maintenance reaches may not be dredged at all. Placement areas that receive dredged material at least once during that five-year period would barely have time to completely recover before being impacted again, while high-use placement areas would never recover completely (presuming they did not become emergent). Continued use of open bay disposal in high maintenance reaches near Port Mansfield in Lower Laguna Madre caused chronic light reduction and has led to extensive losses of seagrasses from waters deeper than 1 m (Onuf, 1994).

Shallow submerged dredged material deposits can revegetate with seagrasses and attract mobile macrofauna to levels comparable with undisturbed seagrasses within five years. Development of comparable benthic communities will take longer. Continued use of these placement areas, especially in high maintenance reaches of the GIWW, means that local primary and secondary productivity (in terms of standing biomass) are continually disrupted at 2 to 10 year intervals. The effects of these local disruptions on Laguna-wide productivity have yet to be measured. Seagrass acreage in Upper and Lower Laguna Madre was estimated to total 730 km^2 in 1988 (Quammen and Onuf, 1993). The total area buried by dredged material at the six sites in this study was estimated to be less than 1 km^2, and whether it was all seagrass habitat prior to deposition is unknown. This study indicated that chronic effects of dredged material deposition on floral and faunal standing crops were limited to the immediate vicinity of each deposit and that recovery was underway within three years. Based solely on areal extent, seagrass burial and the subsequent lessening of Laguna-wide productivity would appear to be small. One indicator of system health that could be examined for trends is a time series of catch rates for fishery organisms, such as those monitored by Texas Parks and Wildlife Department (McEachron and Fuls, 1996). However, such analyses are beyond the scope of this project.

The problem at hand is what to do with dredged material created by maintaining navigation through the Gulf Intracoastal Waterway. Many placement areas are now characterized by emergent islands that were formed by the placement of virgin dredged material with a high sand content during construction of the GIWW (Chaney et al., 1978; Morton et al., 1998). Placement of new maintenance materials on these islands would remove material from the system, particularly if containment levees are employed. Continued deposition of maintenance dredged material in shallow submerged sections near these islands could result in placement areas becoming emergent or too shallow to permit seagrass colonization. Since maintenance material has a lower sand content than the original virgin material, shoaling is most likely to occur where deposits are placed in protected areas (e.g., leeward sides of islands) or if subtidal containment structures are employed (Morton et al., 1998). Placement in exposed or deeper waters leads to sediment re-working and transport away from the site of deposit (Onuf, 1994). Shoaling would result in the permanent loss of seagrass beds and fishery productivity, albeit in exchange for intertidal flats, low islands, or uplands which support other organisms such as algae or birds.

Use of deep, submerged placement areas that are below the depth limits for seagrasses is another option for material placement. In areas of strong wind-induced currents and subsequently continuous high turbidity, use of such deep placement areas could result in reworking of materials and possible loss of nearby seagrass beds. Careful site selection in conjunction with hydrodynamic modeling could locate deep water placement areas where redistribution of dredged materials is minimized. Hydrodynamic modeling could also predict sites where deep water placement and local circulation patterns would permit a build-up of dredged material to elevations in the photic zone and thus increase the chance of seagrass colonization (Rickner, 1979).

The only way to ensure permanent protection of the high primary and secondary productivity of seagrass beds in Laguna Madre from acute and chronic effects of maintenance dredging, while ensuring navigation capability, is to place dredged materials where they cannot be redistributed by wind-induced waves and currents. This means that sediments must be removed from the shallow waters of the ecosystem, either by (1) deposition on previously constructed islands or in deep waters remote from seagrasses, (2) containment at emergent or submerged levee sites within or adjacent to Laguna Madre, or (3) offshore or upland disposal. These recommendations are not new (Rickner, 1979; Onuf, 1994), but it may be time to employ them. Such actions should also decrease the frequency and severity of maintenance dredging, initiate long-term decreases in system turbidity and increases in seagrass acreage, and return ecosystem productivity to a more undisturbed state.

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